

# Alleviating impacts of anthropogenic activities by traditional conservation measures: can a small reef reserve be sustainably managed?

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## Abstract

Zoning schemes that are becoming an important management tool in large marine reserves, are difficult to implement in small reef areas. At the 3.4 km long reef of Eilat (Red Sea), a small (ca. 350 m of coastline) enclosure strategy has been enforced since 1992, while the remaining reef was left open to intense human activities. Here we have investigated for 2.5 y three populations of the branching coral *Stylophora pistillata* (3605 colonies) in a locality within the enclosed area (site NR) and in two areas open to the public, by tossing random quadrats at the shallow lagoonal zone (0.5–1.5 m depth). In the two open sites we found significantly higher levels of colony breakage (14–34% vs. 4–9% in the enclosed site), lower partial mortality levels of colonies (in the first 1.5 y; 7–9% vs. 23–30% at NR), higher recruitment (up to 3.0 vs. up to 0.9 colonies/m<sup>2</sup>), 50% reduction in coral life span (10 vs. 20 y) and an estimated extinction period of 9–10 y for new cohorts as compared to >20 y in the enclosed site. Average colony size and maximal colony size were about half in the open sites. Live coverage fluctuated widely in all sites but was 3 times higher in the enclosed area (1.0–3.0% vs. 0.3–1.1%). Log-transformed size frequency distributions revealed, at the open sites, a shift from small towards medium-size classes and at the enclosed site, a shift from larger to medium size classes. We conclude that the enclosure of a limited core zone, although improved some ecological parameters, was not sufficient to compensate for stress imposed by anthropogenic activities. It is suggested to employ active restoration approaches, such as the “gardening concept”, as supplementary management tools.

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## 1. Introduction

The practice of proclaiming marine ecosystems as marine protected areas (MPAs) and marine reserves are fundamental legislative steps in the conservation of marine habitats (Kelleher and Kenchington, 1992; Gubbay, 1995; Kelleher, 1996; Barr and Thornton, 1998; Tuya et al., 2000). Many of the model MPAs consist of large ecosystems, encompassing a variety of

coastal and oceanic habitats. Best known example is the Great Barrier Reef Marine Park in Australia (GBRMP). In the GBRMP, the world's largest MPA, a zoning scheme has been successfully applied, with three major zones of management: conservation areas (core zones), permitted use areas (National Park zones), and general use zones. In other MPAs, highly sensitive grounds, e.g., nurseries and spawning sites, receive additional special protection measures such as the adaptation of no-use legislation (Christensen et al., 1996) or consideration of connectivity and larval capabilities to migrate geographic distances far greater than the protected area size (Ogden, 1997; Roberts, 1997; Allison et al., 1998). In

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large MPAs, different zoning systems surrounded by limited access areas and/or by open access areas, are employed as primary management tools for the protection of unique habitats, resolving social/economic vis-a-vis protection conflicts of interests.

Small reef areas put forward additional management challenges and require more intensified management activities (Schwartz, 1999). They may encompass insufficient core zones to ensure sustainable management. Thus, in contrast to large reserves, where additional ‘belts’ of managed zones are usually set up to protect the core area (Noss and Harris, 1986), small reef areas are often exposed directly to various antropogenic activities without being sheltered by buffer zones. In a few cases (e.g., White and Vogt, 2000), small reef areas are claimed to be successfully managed. In many others, however, such as the Coral Nature Reserve in Eilat, Northern Red Sea, and the Hikkaduwa Marine Sanctuary in Colombo, Sri Lanka (two small-size MPAs of merely 4 km coastline, each), rapid degradation has been documented (Loya, 1990; Riegel and Velimirov, 1991; Fishelson, 1995; De-Silva, 1997a,b; White et al., 1997; Epstein et al., 1999; Zakai and Chadwick-Furman, 2002; Wielgus et al., 2003, 2004) despite the existing tough management practices.

The reefs in the Coral Reef Reserve of Eilat, Israel, were considered as the most diverse reefs in the world (Loya, 1972). Decades of various anthropogenic activities have resulted in continuous reef degradation (Loya, 1990; Fishelson, 1995). More recently, hotel development and tourist activities, have transpired as being a major agent of reef decline in Eilat (Riegel and Velimirov, 1991; Wilhelmsson et al., 1998; Epstein et al., 1999; Zakai and Chadwick-Furman, 2002; Bongiorno et al., 2003), confirming outcomes from other reef sites worldwide (Talge, 1992; Allison, 1996; Chadwick-Furman, 1997; Muthiga and McClanahan, 1997; Ormond et al., 1997; Roupheal and Inglis, 1997). In 1992 a “no-use” zone policy was initiated within this small MPA on a restricted area of ca. 350 m long reef (Meshi, pers. comm.). As in other types of marine habitats (e.g., Laffoley, 1995; Brailovskaya, 1998; Lauck et al., 1998), such a small coral reef enclosure system has not often been subjected to theoretical nor to practical evaluations.

Six years later, we began to monitor (Epstein et al., 1999) the effectiveness of this measure. We investigated demographic properties of coral populations within the ‘closed-to-the-public’ area (close site), comparing it to two adjacent areas that were open-to-the-public (open sites). The major aim of this research was to follow populations of *Stylophora pistillata*, one of the most common coral species in the northern Gulf of Eilat, for the purpose of elucidating the effectiveness of the employed management actions. Hereby, we present results of a 2.5 year follow-up study designated to assess the

impact of a non-use management policy implemented at Eilat’s Coral Nature Reserve, >10 years ago.

## 2. Material and methods

### 2.1. Study area

The fringing coral reef of Eilat is located at the northwestern tip of the Gulf of Eilat and is the most northern extension of the Red Sea reef system (Fishelson, 1995). This reef encompasses two differently managed zones (Fig. 1); the northerly-situated “Hof Almog” Reserve (ca. 1.4 km of coastal area), where a 1.1 km strip is fenced and number of visitors is strictly regulated, and the southern “Hof Dromi” Reserve (ca. 2 km long) that stretches to the south, towards the Israeli-Egyptian Taba border crossing, and is regulated as an open site nature reserve (Ortal and Nemtsov, 1997).

Here we studied the shallow lagoonal ‘trampling zone’ (0.5–1.5 m depth) at three reef sites (Fig. 1). Two open sites are situated at the Hof Dromi Nature Reserve: (1) the reef in front of the Princess Hotel (site HP), opposit the Taba border crossing; (2) the reef in front of the InterUniversity Marine Laboratory (site MBL), located 1 km north of HP. In both sites, access to deep water is possible via walkways, although entrance through the lagoon is also allowed. The third site, the no-use lagoon zone in Eilat, which is found within the strictly protected Hof Almog Reserve (site NR), 0.5 km north of the MBL and is forbidden to all recreational activities. Research activities are limited only to snorkeling during high tide. Diving and swimming are allowed only beyond the reef wall and the entrance to the deep reef is strictly confined to walkways.

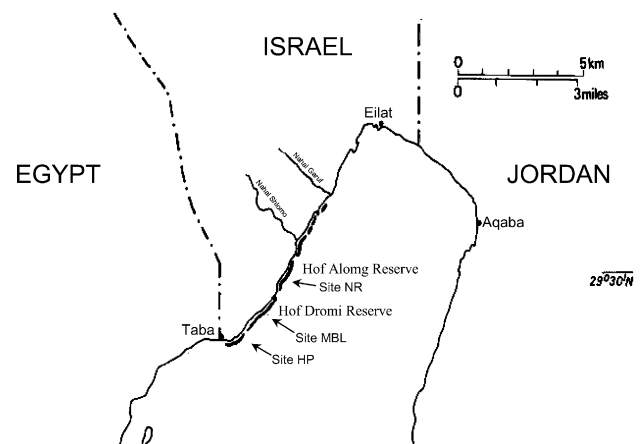


Fig. 1. The northern tip of the Red Sea Gulf. The three study sites, HP (Hotel Princess), MBL (Marine Biology Lab), and NR (Nature Reserve) are indicated with arrows.

## 2.2. Studied species

*Stylophora pistillata*, a branching Indo-Pacific coral species, is one of the most abundant coral species in Eilat, dominating the lagoon, rear-reef, and reef-flat coral communities, and also occurring in abundance at the deeper parts of the fore-reef and the slope (Loya, 1972). It is a species of focal importance in the ecology of Eilat's reef system, notable for its biological characteristics such as high recruitment rates, rapid growth and development, fast regeneration, early onset of reproduction and long reproductive season, the high number of fish and invertebrates residing between and above the colony branches, and its major contribution as a reef framework builder (Loya, 1976a,b; Rinkevich and Loya, 1979a,b, 1987; Loya, 1985; Muscatine et al., 1985; Rinkevich et al., 1991). Its branching colony morphology renders this species vulnerable to breakage. It is therefore a suitable subject for this follow-up study.

## 2.3. Size frequency distribution

This study was conducted from May 1998 to October 2000. Each site was sampled six times (twice a year) during the months of April, May and October. At each site, a total area of ca. 400 m<sup>2</sup> of the trampling zone was repetitively sampled, using 1 m<sup>2</sup> randomly selected quadrats, a protocol that enables us to apply the results to the whole site (see Epstein et al., 1999).

Within each quadrat, height, length, and perpendicular width of all *S. pistillata* colonies were measured using a plastic caliper (accuracy 1.0 mm). The geometric mean radius (GMR,  $\bar{r}$ ) was calculated following Loya (1976a). "Breakage" was defined as missing or broken branches within a colony in 20% bins. Similarly, "partial tissue loss" was defined as dead parts on existing branches within a colony estimated in 20% bins. To evaluate extent of breakage and partial mortality in different size categories, we divided the colonies into five GMR size classes of <1.1, 1.1–2.0, 2.1–4.0, 4.1–10 and >10.1 cm that reflect different stages in onset of colony development (Loya, 1976a,b,c; Rinkevich and Loya, 1979a,b, 1987). For linear size frequency distribution analyses, the colonies were divided into size classes of 1 cm GMR intervals, as the average yearly growth rate of this species had been estimated to be ca. 1 cm GMR (Loya, 1976c; Muscatine et al., 1985).

To further describe demographic changes in *S. pistillata* populations through time, we analyzed the colony size frequency of each population by log transforming the size data (Bak and Meesters, 1998). This method is based on the relative changes in size (Vermeij and Bak, 2003), which compensate for the large differences in absolute colony growth between small and large colonies over the same time period. This method has been particularly recommended to elucidate

structure alterations in populations of degraded and marginal reefs (Bak and Meesters, 1999; Meesters et al., 2001). The resulting high-resolution size-frequency distribution better reveals life-history processes of recruitment, juvenile mortality, and partial colony death, particularly due to an increase in the number of small-size classes.

Here, we used a log-transformation with base 2 of the original size data ( $\ln \{size\} / \ln \{2\}$ ; Vermeij and Bak, 2003). We arbitrarily defined successive size classes by choosing an iteration step over which the size increase was calculated. We chose an iteration step of 0.3 ( $2^{-1}, 2^{-0.7}, 2^{-0.4}, 2^{-0.1}, 2^{0.2}, 2^{0.5}, \dots$ ) that yielded 22 size classes of 0.27, 0.33, 0.41, 0.50, 0.62, 0.76, 0.93, 1.15, 1.41, 1.74, 2.14, 2.64, 3.25, 4.00, 4.92, 6.06, 7.46, 9.19, 11.31, 13.93, 17.15, and 21.11 cm GMR, respectively. This detailed size class distribution enables us to follow and compare small changes in size-frequency distributions.

## 3. Results

For each census of the six sampling periods, the quadrats covered between 8% and 24% of the studied area, at each site (Table 1). Measurements were taken from 3605 *S. pistillata* colonies; 1165 at HP, 1250 at MBL and 1190 at NR in 308, 277 and 331 quadrats, respectively. Average number of colonies per 1 m<sup>2</sup> was not statistically different between the 3 sites (one way Anova,  $p > 0.05$ ), and ranged from 3.6–5.9 colonies/m<sup>2</sup> at the open sites to 2.9–5.0 colonies/m<sup>2</sup> at NR. The total combined average colony size at HP and MBL was  $1.9 \pm 1.8$  cm GMR, about half from the protected NR site ( $4.0 \pm 3.9$  cm GMR). Maximal colony size reached 9.9 cm GMR at the open sites, less than a half from the NR (22.4 cm GMR). The calculated total percentage live cover of *S. pistillata* populations fluctuated at all studied sites, but was almost three times higher at the protected NR site (1.0–3.0%) as compared to 0.3–1.1% at the open sites (Duncan's multiple range,  $p < 0.05$ ; Table 1).

### 3.1. Linear size frequency distribution

*Stylophora pistillata* populations at all sites were characterized by high proportions of small-size colonies (Fig. 2). The two smallest size classes (0.5–1 and 1.1–2.0 cm GMR; corresponding to ages  $\leq 2$  years) comprised the majority of the populations, up to 75% at the open sites (MBL; April 1999) and 46% at the NR (October 1999; Fig. 2). Larger colonies were more abundant at the NR site than in the open sites (Duncan's multiple range,  $p < 0.05$ ). Fig. 2 depicts only colony size distributions of up to 10.0 cm GMR, since individuals larger than this size were not found at the open sites. At these locations,

Table 1

*Stylophora pistillata* populations: summary of the six censuses at the HP, MBL, and MR sites during the 2.5 y study

Site	Date	No. quadrats	Total sampled area (%)	Average no. of corals/quadrat ( $\bar{X} \pm \text{SD}$ )	Total no. of colonies	Average colony GMR ( $\bar{X} \pm \text{SD}$ ; cm)	<i>S. pistillata</i> coverage area (%)
HP	May 98	48	12.0	4.1 $\pm$ 2.4	175	2.2 $\pm$ 1.9	0.60
	October 98	49	12.0	3.6 $\pm$ 2.2	171	1.7 $\pm$ 1.6	0.30
	May 99	96	24.0	3.4 $\pm$ 2.7	327	1.7 $\pm$ 1.8	0.31
	October 99	49	12.0	3.9 $\pm$ 2.7	186	2.4 $\pm$ 2.0	0.68
	May 00	34	8.5	4.8 $\pm$ 2.5	163	2.7 $\pm$ 1.8	1.10
	October 00	32	8.0	4.5 $\pm$ 2.2	143	2.3 $\pm$ 1.8	0.71
MBL	May 98	51	13.0	4.2 $\pm$ 3.5	214	1.8 $\pm$ 1.8	0.42
	October 98	48	12.0	4.5 $\pm$ 3.2	211	1.6 $\pm$ 1.6	0.36
	May 99	77	19.0	4.0 $\pm$ 3.0	312	1.7 $\pm$ 1.7	0.36
	October 99	31	8.0	5.9 $\pm$ 4.1	183	2.2 $\pm$ 2.1	0.89
	May 00	36	9.0	4.6 $\pm$ 2.8	168	1.7 $\pm$ 1.4	0.42
	October 00	34	8.5	4.6 $\pm$ 2.8	162	2.4 $\pm$ 1.7	0.83
NR	May 98	69	17.0	3.5 $\pm$ 2.6	242	3.9 $\pm$ 3.4	1.70
	October 98	76	19.0	2.9 $\pm$ 1.8	217	3.4 $\pm$ 3.2	1.00
	May 99	75	19.0	3.8 $\pm$ 1.9	287	5.0 $\pm$ 4.4	3.00
	October 99	32	8.0	5.0 $\pm$ 2.3	159	3.7 $\pm$ 4.0	2.20
	May 00	46	11.5	3.3 $\pm$ 2.3	151	4.7 $\pm$ 4.9	2.30
	October 00	33	8.0	4.2 $\pm$ 2.9	134	3.8 $\pm$ 3.7	2.40

the abundance of GMR = 6.1–7.0 cm colonies dropped to ~3% (up to 8% at NR) and the larger sized classes (up to 10.0 cm GMR) decreased to 1%, or were not represented in some censuses (up to 6% at NR). The 10.1–11.0-size class in NR was relatively abundant (4–5% of colonies) and size classes up to the maximal size of 22.4 cm represented 1% of colonies (Duncan's multiple range test  $p < 0.05$ ).

### 3.2. Log transformed size distribution

Fig. 3 depicts colony size frequencies in each *S. pistillata* population, at the beginning (May 1998) and at the end of the monitoring period (October 2000). The size frequencies of the colonies have shifted over this period. After 2.5 years, the distribution of all size classes up to 1.15 cm (recruitment classes combined), decreased from May 1998 to October 2000 by 14% at the HP, 41% at the MBL, and 13% at the NR site. Distribution of the combined classes up to 0.50 cm decreased by 25% at the HP (26.9–20.3%), 46% at the MBL (29.9–16.2%) and by 14% at the NR (18.2–15.7%). This decrease was further noted in the distribution of the combined 0.50–1.15 cm classes at the MBL site where a 35% decrease in abundance was documented (from 21.1% on May 1998 to 13.8% on October 2000). An increase in the intermediate size classes distribution was recorded after 2.5 years. The size classes 2.64, 3.25 and 4.00 cm increased at the HP site by 40–83%, at the MBL site by 46–135%, and 80–116% at NR site. However, at the NR site this was accompanied by a decrease in the larger classes 6.06–9.19 cm abundance by 37–57%.

After 2.5 years, therefore, recruitment (colonies up to 1.15 cm) decreased at all three studied sites with the largest drop of 41% at the MBL. Intermediate size classes increased in proportion at all sites, but the larger classes at the protected NR site decreased.

### 3.3. Partial colony breakage

During the 2.5 year study period, a significantly higher numbers of partial colony breakage was recorded in the HP and MBL sites (ranging 15–34% and 14–27% of the populations, respectively; Anova,  $p < 0.05$ ) than in the protected NR (4–9%, Fig. 4(a)). An unusually high level of 34% and 27% broken colonies were documented on May 1998 at both open to the public sites (Fig. 4(a)) as compared to only 8% in NR site. The other five censuses showed reduced breakage frequencies.

This outcome is further reflected in the within group-size damage analyses (Table 2), where the distribution of broken colonies increased in larger colonies at all three sites. In the open sites, 34–79% of the 4.1–10.0 cm group size colonies were partially damaged as compared to 0–5% in the <1.1 cm group size (Table 2). There were, nevertheless, considerable differences between NR and the open sites. While frequencies of broken colonies in the smallest size class (<1.1 cm) were low and similar in all sites (0–5%), in the open sites they were augmented in the second size class (1.1–2.0 cm; May 1998) to 30% at the HP and MBL as compared to 3–12% in the NR (Duncan's multiple range test,  $p < 0.05$ ). Whereas, at the open sites, frequencies of broken colonies further increased in the larger size classes, they remained

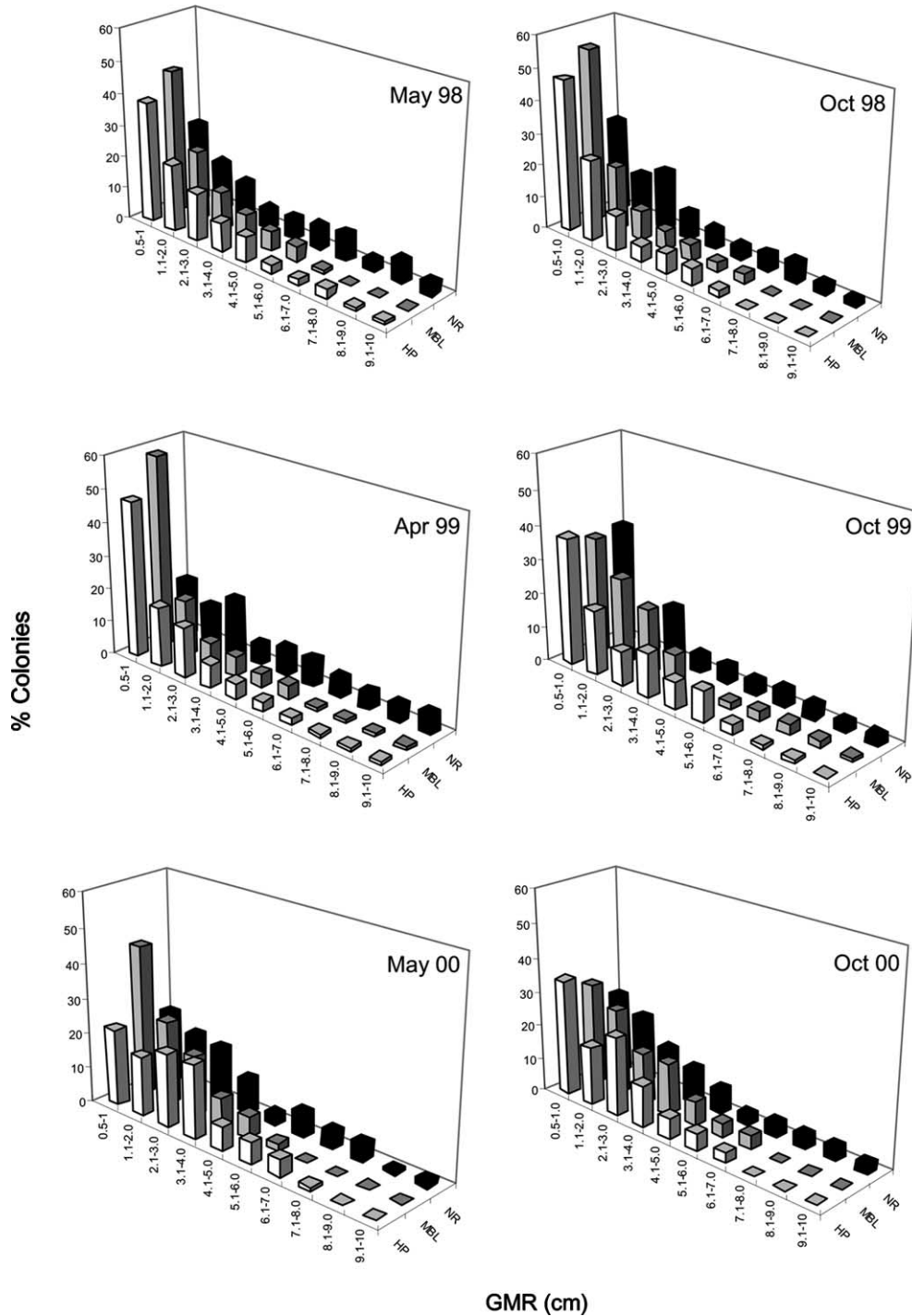


Fig. 2. One centimeter interval size frequency distributions in of *S. pistillata* populations at the trampling zones of the 3 sites during the 2½ year study. Only the first 10 classes are given since larger colonies were not found at the HP and MBL.

relatively low in the protected NR, except for the 10.1 cm GMR group (peaked at 45%).

When evaluating branch breakage at the level of the individual colony (Table 2), in the open sites, usually >30% of the damaged colony, on the average, was broken (reaching values of >50% colony damage). In the NR the figures fluctuated but were generally lower. Even in the largest size class ( $\geq 10.1$  cm GMR) the

maximal average value of damaged parts within a single colony was only 30%.

### 3.4. Partial colony mortality

The NR population exhibited significantly higher (Anova,  $p < 0.05$ ) levels of Partial colony mortalities (PCMs) (23–30%) than the open sites (up to 12%) during

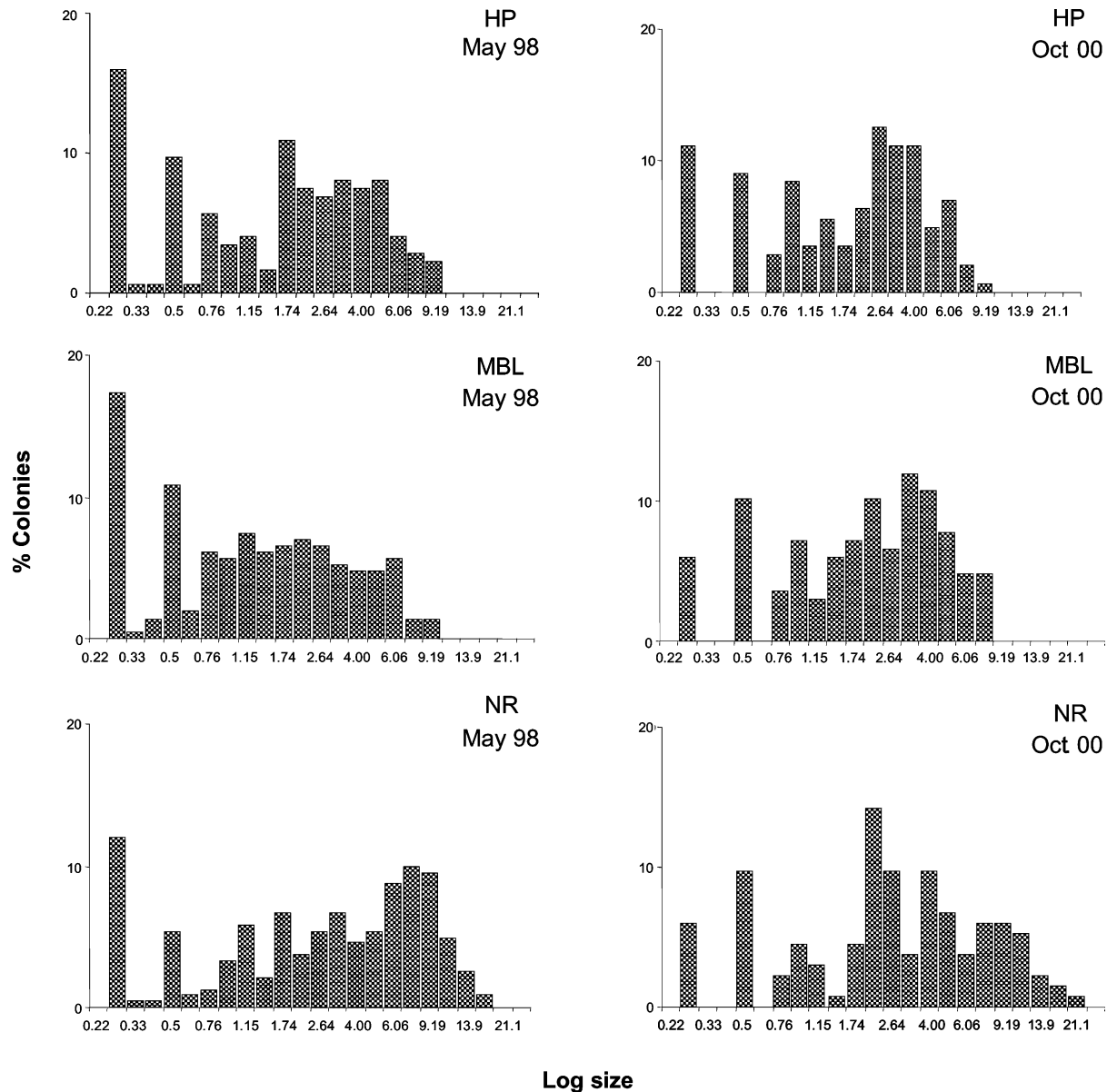


Fig. 3. *Stylophora pistillata* populations high-resolution (log-transformed) size-class frequency distribution histograms at the three studied reef sites on May 1998 and October 2000.

the first 1.5 years of the study (Fig. 4(b)), levels that persisted for the rest of the studied period (24–28%). In the two open sites, a sharp increase in the PCM was documented in October 1999 (17% at HP and 20% at MBL), and from May 2000, similar levels of PCMs were recorded in all three sites (Fig. 4(b)). With the increase in colony size, more colonies were affected at all sites (e.g., 21% and 18% of the colonies in the first two size classes vs. 69% in the 4.1–10.0 cm GMR class at the MBL, May 2000 census; Table 3). In the open sites, the average size of the impaired colonies ( $3.3 \pm 0.6$  cm GMR) was about half of the NR ( $6.9 \pm 1.5$  cm; Duncan's multiple range test,  $p < 0.05$ ).

At the colony level, PCM signaled a state of colony degradation that leads to colony demise. The smallest size class displayed ca. 75% and 95% PCM in the open and NR sites, respectively, and in the 4.1–10.0 cm size class PCM levels were 23–80% (Table 3). The PCM augmentation with time at the population level (Fig. 4(b)) was also reflected at the colony level. For example, in the 4.1–10.0 cm size class, it increased in all sites from 23–34% PCM on the average in May 1998 to ca. 80% in May 2000 (Table 3). In the <1.1 cm and the 1.1–2.0 cm classes we found many dead colonies (were not included in the analyses). Therefore, the '0' PCM values for small colonies do not reflect healthy

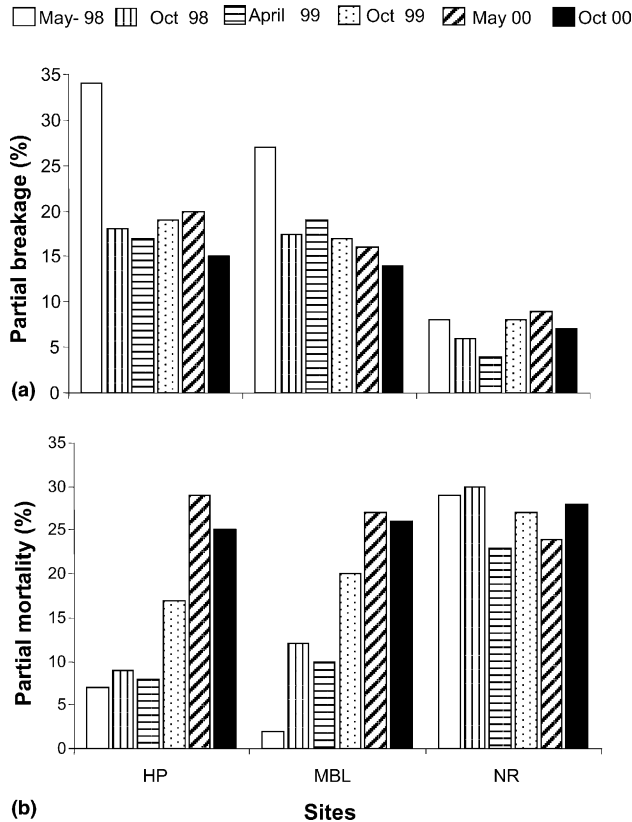


Fig. 4. Impairment rates of the entire *S. pistillata* population (May 1998 to October 2000) at the three study areas, measured by: (a) % partial colony breakage; (b) % partial colony mortality.

populations. Small colonies, being more vulnerable, die faster, curtailing the intermediate PCM phase.

### 3.5. Recruitment and survivorship: cohort analyses

We analyzed the fate of cohorts based on *S. pistillata* reproductive seasonality (Rinkevich and Loya, 1979a,b, 1987) and colony growth rates. Colonies of GMR < 1.1 cm size class found in each May census represented the recruitment cohort of the previous year. May census analyses (3 consecutive reproductive seasons, May 1998–2000) allowed us to follow the fate of year cohorts as follows: Colonies of the GMR < 1.1 cm class of May 2000 census were considered as last year's (1999) recruitment and were analyzed only once. The 1998 cohort was observed twice, in May 1999 (<1.1 cm class) and in May 2000 (1.1–2.0 cm class). The year 1997 cohort occurred in three size classes, as the <1.1 cm group of May 1998 census, the 1.1–2.0 cm class of May 1999 census, and as 2.1–3.0 cm size class of May 2000 census. Following that rationale, the fate of previous years cohorts could be extrapolated back, not only to year 1992, when the enclosure was initiated, but also to year 1987 (encompassed the group size 10.1–11.0 cm GMR; Fig. 5). Some coral colonies may be “shifted” from their actual

size classes to “younger” classes resulting from partial colony breakage events (Hughes and Jackson, 1980; Bak and Meesters, 1998). It was impossible to validate these numbers.

At all three sites, the number of colonies per cohort had gradually decreased with increasing colony size (Fig. 5). The 1992 cohort (the year when the enclosure was initiated) was present in all NR May censuses (0.2 colonies/m<sup>2</sup> in the 5.1–6.0, 6.1–7.0, and 7.1–8.0 cm group size, respectively). This cohort was reduced to 0.1 colonies/m<sup>2</sup> at the open sites and was absent from the 7.1–8.0 cm class in the MBL May 2000 census. The 1987–1991 cohorts had disappeared in the last census (May 2000) from both open sites but still found at levels of 0.05–0.2 colonies/m<sup>2</sup> at NR. Survivorship in the protected NR was higher (colonies of the age of 21 years were still alive) than in the open sites where it took about 9–10 years for cohort extinction. The 1992 cohort was, therefore, prone to extinction at the open sites in the years 2001–2002. While survivorship at the NR site has improved, its recruitment rates had been below the values of the HP and MBL. The 1997–1999 recruitment values in NR were 0.6–0.9 colonies/m<sup>2</sup> as compared to 1.0–1.6 colonies/m<sup>2</sup> in HP and 1.9–3.0 colonies/m<sup>2</sup> in MBL site.

## 4. Discussion

Evaluation of the effectiveness of measures implemented in marine protected areas is an essential step in developing of sustainable conservation strategies (McNeill, 1994; Stanford and Poole, 1996; Jackson, 1997; Allison et al., 1998; Brailovskaya, 1998; Done and Reichelt, 1998; Fluharty, 2000; Keough and Quinn, 2000; Nickerson-Tietze, 2000; Rose, 2000). Here we have followed the effectiveness of the declared limited ‘no use zone’ policy implemented in a small MPA by studying three *S. pistillata* shallow water populations for 2.5 y.

We found that *S. pistillata* populations at the open sites exhibited significantly higher levels of coral breakage, and a reduction in maximal colony size, that implies faster extinction rates as compared with NR site. As to partial colony mortality levels, initially, only site NR displayed high levels, but as of May 1998, increased levels were recorded also at the two open site populations. Coral populations at all sites displayed large temporal fluctuations in living area coverage and also a reduction in recruitment. Decisive conclusions pertaining to causes and pathways of these fluctuations cannot be reached at this stage because of our scanty knowledge. However, the data obtained here may be used as an important tool for the evaluation of the conservation strategy implemented at the Eilat reef reserve. This nature reserve, despite all direct conservation measures, is

Table 2

Percentages of broken colonies at the different size classes and colony level ( $\bar{x} \pm SD$ ) in the three studied sites over the six censuses (SC = size class)

Site	Date	SC < 1.1; breakage at the		SC = 1.1–2.0; breakage at the		SC = 2.1–4.0; breakage at the		SC = 4.1–10.0; breakage at the		SC $\geq$ 10.1; breakage at the	
		Population level	Colony level	Population level	Colony level	Population level	Colony level	Population level	Colony level	Population level	Colony level
HP	May 98	3	50 $\pm$ 0	30	52 $\pm$ 32	62	38 $\pm$ 22	79	28 $\pm$ 2	–	–
	October 98	1	30 $\pm$ 0	15	53 $\pm$ 25	33	27 $\pm$ 10	75	45 $\pm$ 20	–	–
	Apr 99	4	43 $\pm$ 22	26	33 $\pm$ 15	33	33 $\pm$ 19	50	39 $\pm$ 9	–	–
	October 99	2	40 $\pm$ 0	20	29 $\pm$ 11	23	33 $\pm$ 13	44	33 $\pm$ 17	–	–
	May 00	5	50 $\pm$ 0	14	35 $\pm$ 13	23	37 $\pm$ 17	34	38 $\pm$ 20	–	–
	October 00	0	0	4	40 $\pm$ 0	22	30 $\pm$ 14	48	46 $\pm$ 18	–	–
MBL	May 98	4	48 $\pm$ 17	21	53 $\pm$ 24	67	48 $\pm$ 22	74	34 $\pm$ 24	–	–
	October 98	3	36 $\pm$ 15	12	40 $\pm$ 12	53	53 $\pm$ 14	61	49 $\pm$ 21	–	–
	Apr 99	4	33 $\pm$ 18	15	24 $\pm$ 9	46	34 $\pm$ 16	73	34 $\pm$ 18	–	–
	October 99	2	30 $\pm$ 0	5	20 $\pm$ 14	30	38 $\pm$ 17	54	49 $\pm$ 19	–	–
	May 00	3	35 $\pm$ 21	10	40 $\pm$ 14	34	43 $\pm$ 20	54	46 $\pm$ 17	–	–
	October 00	0	0	8	27 $\pm$ 6	18	29 $\pm$ 14	41	40 $\pm$ 21	–	–
NR	May 98	2	30 $\pm$ 0	3	10 $\pm$ 0	7	20 $\pm$ 18	15	22 $\pm$ 15	31	25 $\pm$ 6
	October 98	0	0	0	0	0	0	12	27 $\pm$ 11	45	20 $\pm$ 7
	Apr 99	2	50 $\pm$ 0	5	15 $\pm$ 7	2	20 $\pm$ 0	6	28 $\pm$ 10	7	25 $\pm$ 7
	October 99	0	0	6	40 $\pm$ 0	0	0	23	27 $\pm$ 14	23	30 $\pm$ 20
	May 00	0	0	12	17 $\pm$ 6	0	0	12	18 $\pm$ 5	29	23 $\pm$ 8
	October 00	0	0	0	0	3	10 $\pm$ 0	18	20 $\pm$ 9	22	15 $\pm$ 7



Table 3

Percentages of partial colony mortality (PCM) at the different size class and colony levels ( $\bar{x} \pm SD$ ) at the 3 studied sites over the 6 censuses (SC = size class)

Site	Date	SC < 1.1; PCM at the		SC = 1.1–2.0; PCM at the		SC = 2.1–4.0; PCM at the		SC = 4.1–10.0; PCM at the		SC $\geq$ 10.1; PCM at the	
		Population level	Colony level	Population level	Colony level	Population level	Colony level	Population level	Colony level	Population level	Colony level
HP	May 98	2	20 $\pm$ 0	3	40 $\pm$ 0	7	43 $\pm$ 6	23	34 $\pm$ 16	–	–
	October 98	1	40 $\pm$ 0	5	25 $\pm$ 7	22	28 $\pm$ 22	30	38 $\pm$ 19	–	–
	April 99	4	90 $\pm$ 15	2	30 $\pm$ 0	22	54 $\pm$ 32	27	41 $\pm$ 32	–	–
	October 99	0	0	9	57 $\pm$ 39	23	70 $\pm$ 39	44	65 $\pm$ 33	–	–
	May 00	0	0	0	0	37	89 $\pm$ 27	35	80 $\pm$ 29	–	–
	October 00	21	95 $\pm$ 10	17	73 $\pm$ 32	26	54 $\pm$ 36	43	57 $\pm$ 30	–	–
MBL	May 98	0	0	0	0	5	25 $\pm$ 21	11	23 $\pm$ 15	–	–
	October 98	1	30 $\pm$ 0	7	43 $\pm$ 6	40	50 $\pm$ 27	44	38 $\pm$ 26	–	–
	April 99	0	0	4	30 $\pm$ 0	16	35 $\pm$ 16	30	33 $\pm$ 23	–	–
	October 99	0	0	9	88 $\pm$ 25	28	64 $\pm$ 36	63	56 $\pm$ 38	–	–
	May 00	21	95 $\pm$ 14	18	57 $\pm$ 34	34	54 $\pm$ 27	69	50 $\pm$ 33	–	–
	October 00	13	97 $\pm$ 8	20	79 $\pm$ 31	35	55 $\pm$ 36	45	57 $\pm$ 35	–	–
NR	May 98	3	40 $\pm$ 4	8	40 $\pm$ 17	33	43 $\pm$ 21	52	30 $\pm$ 24	46	23 $\pm$ 16
	October 98	3	75 $\pm$ 21	13	23 $\pm$ 10	27	36 $\pm$ 22	65	39 $\pm$ 22	73	28 $\pm$ 14
	April 99	0	0	8	63 $\pm$ 35	27	54 $\pm$ 36	38	64 $\pm$ 34	50	65 $\pm$ 31
	October 99	0	0	18	53 $\pm$ 25	25	59 $\pm$ 38	65	55 $\pm$ 36	46	33 $\pm$ 31
	May 00	0	0	4	40 $\pm$ 0	23	53 $\pm$ 37	38	43 $\pm$ 34	67	44 $\pm$ 26
	October 00	0	0	22	88 $\pm$ 20	27	75 $\pm$ 30	58	64 $\pm$ 35	22	20 $\pm$ 14

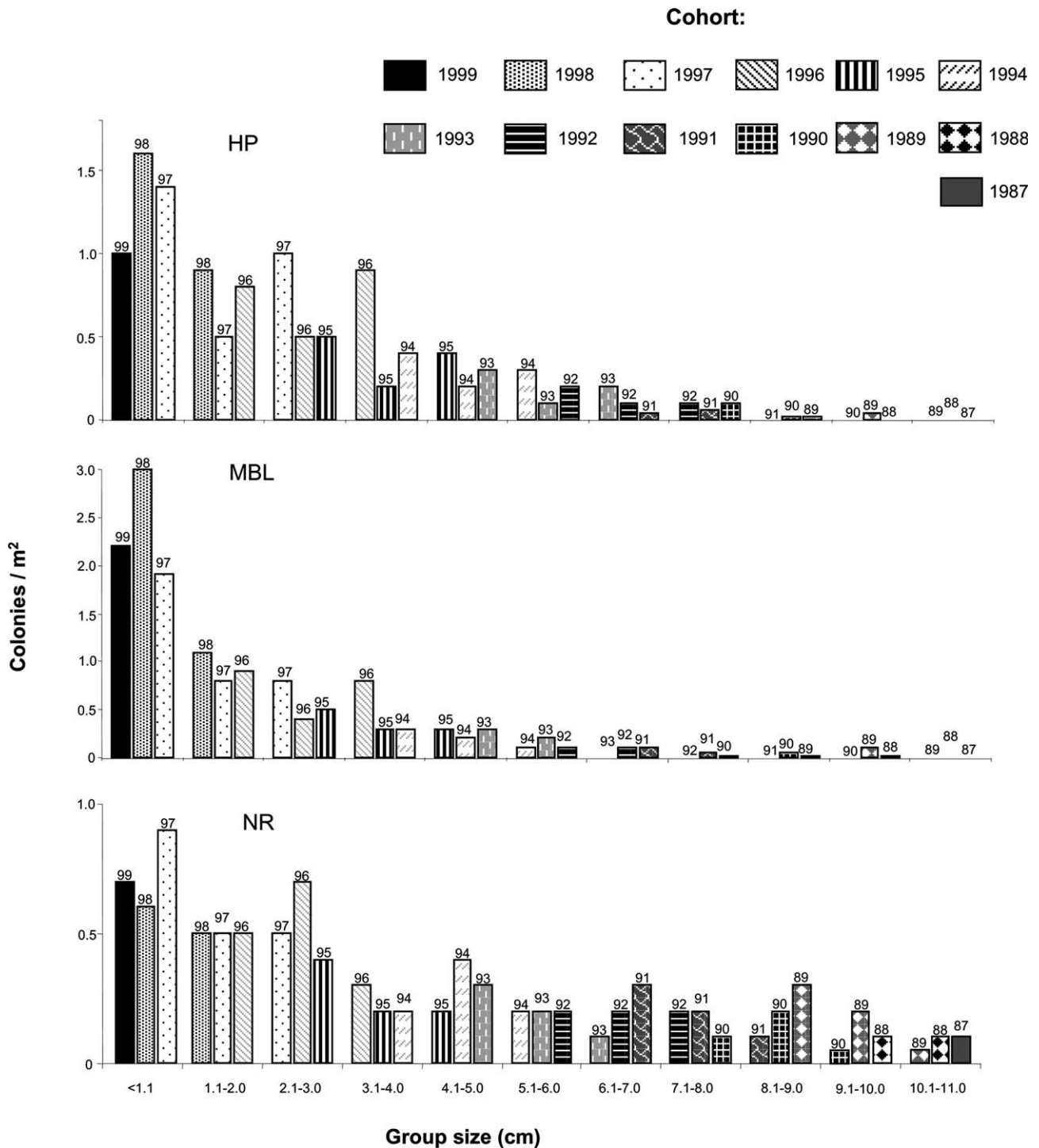


Fig. 5. A follow-up analysis (May 1998–2000) of 13 predicted *S. pistillata* cohorts from the three study areas. Numbers above columns denote year of settlement, i.e., the 1999 cohort of the <1.1 size class was measured on May 2000, etc.

still being impacted by antropogenic agents originating outside the reserve. Possible sources are the excessive coastal, municipal and agricultural development in the area as well as sewage spillages and water entrophication (Bongiorni et al., 2003; Wielgus et al., 2004). Moreover, given the patched nature of recruitment at Eilat's reef in both, space and time (Epstein et al., 1999;

Glassom, 2002), a three year survey of recruits may not be sufficient to provide the full spectrum of this phenomenon.

*Stylophora pistillata* is a fairly small-size, short living coral species (Loya, 1976a). Populations of such a species are generally characterized by negatively-skewed size frequency distributions, with many small and few

large colonies (Babcock, 1991; Soong, 1993; Bak and Meesters, 1998, 1999; Meesters et al., 2001). However, after 2.5 years of observations we documented shifts in size frequency distributions towards the intermediate classes at all three sites, and a reduction in the abundance of larger colonies at the NR site. These shifts in the protected area, further amplified by the high PCM levels, contradicted the expected outcomes of the 1992 ‘no-use’ zone policy. This policy and the implementation of other traditional management measures in the Eilat reef diving and snorkel restrictions, establishment of walkways and pathways for visitors in the reef, collection permits, pollution control strategies, although improving the NR situation as compared to the open sites, are insufficient to hold coral populations in a long-term equilibrium state.

Ecological stability can be viewed in several ways, such as the ecosystem ability to resist forces that act to impose a change on it (‘resistance stability’, Sutherland, 1974), or the time elapsed to return to a pre-disturbance state (‘resilience’, Margalef, 1968). Holling (2001) further suggested that resilience should be viewed as the magnitude of disturbance that a system can absorb without the alteration of its variables. Our findings reflect therefore, both, reduced resistance-stability and ecosystem-resilience in Eilat’s reefs, including the protected NR site. This conclusion is further supported by unrelated long-term studies on coral community structure in the northern Red Sea (Loya, 1972, 1975, 1990; Fishelson, 1973, 1995; Wielgus et al., 2003), where the poor resilience of reef communities and deterioration due to human perturbations were confirmed. Loss of the larger *S. pistillata* colonies at the open sites may reduce reproductive output and implies a serious population decline (Hughes and Tanner, 2000). The decreased reproductive activity within the protected NR site may eventually affect the southerly situated *S. pistillata* populations. Furthermore, there are no large coral populations upstream, north to the NR site that may accommodate the loss of coral colonies in the nature reserve by supplying large numbers of recruits through planulae production. These outcomes contradict Glasom (2002) results that recorded (year 2001) high densities of juvenile corals in several localities along Eilat. Higher recruitment patterns was also recorded on artificial reefs at Eilat, colonized within 100 months with >30 coral species (Abelson and Shlesinger, 2002).

The small-scale enclosure policy in Eilat, however, is ineffective in ameliorating long-term impairments such as expected of similar policy implemented in large reef areas (see also Risk, 1999). The traditional conservation approaches carry, therefore, default merits as sustainable tools for small coral reef management. In Eilat, increasing impacts of human activities from reef based tourism appear to be the most important destructive agents (reviewed in Hawkins and Roberts, 1994;

Rinkevich, 1995; Zakai and Chadwick-Furman, 2002; Wielgus et al., 2004). This reef is suffering from “the highest known frequency of recreational diving on a small reef area anywhere in the world” (Zakai and Chadwick-Furman, 2002). Over the last decade, more than 1 million people per annum have visited Eilat (Fishelson, 1995). When surveying the behaviour of SCUBA divers on the coral reefs of Eilat based on 1 hour average diving time, Zakai and Chadwick-Furman (2002) found that ca. 400,000 coral colonies per year are being broken. In addition, divers activities resulted in >2 million cases per year of sediment deployment or fin contacts, >1.4 million cases per year of hand contacts, and about 1 million cases per year of coral-SCUBA gear contacts. This impact on the reef at Eilat “is ecologically unsustainable” (Zakai and Chadwick-Furman, 2002). Tourism is described as merely one facet of anthropogenic impact. Local agriculture, rampant coastal and municipal development, sewage spillage, eutrophication, and many other human activities contribute to the deterioration of the reef environment.

Since no buffer system can be applied to this small reserve, active, site-specific restoration measures are necessary to rehabilitate damages. Moreover, the failure to enhance reef growth by traditional approaches reveals that, at this stage, efforts should be concentrated less on how to conserve what is left and rather actively be engaged in how to restore (Young, 2000). Active restoration strategies such as the ‘Reef Gardening’ concept (Rinkevich, 1995, 2000; Epstein et al., 2001, 2003) can rescue reefs from the wider, on-going impacts of human activities and be used as a supplementary management approach for rehabilitation of small, exploited and deteriorating MPA’s. By using these strategies, new colonies of branching species like *S. pistillata* are formed in less than 1 year (Epstein and Rinkevich, 2001; Bongiorno et al., 2003) circumventing the apparent reduction in natural recruitment. These colonies may go, within a short period after their transplantation, through sexual maturation with high survivorship rates, and attain a size that may potentially contribute to the reef resistance.

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